Spatiotemporal variation in the long-term fish assemblages of Buck Creek, Delaware County, Indiana

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Abstract

Buck Creek is a spring-fed, cool-water tributary of the West Fork White River, Indiana. The Muncie Bureau of Water Quality sampled fishes and monitored water temperature in Buck Creek annually from 1986-2018. For this study, we utilized long-term fish data from the Bureau of Water Quality to evaluate spatial and temporal changes in the fish assemblages of Buck Creek in Delaware County, Indiana, USA. Non-metric multidimensional scaling (NMDS) was used to describe changes in the fish assemblages over space and time. Linear mixed effects models were used to evaluate the relationship between environmental factors and the fish assemblages. The spatial NMDS results were separated in distinct groups of upstream and downstream assemblages. This was characterized by a shift of headwater specialists shifting to large-river species. The temporal NMDS results were separated into distinct annual assemblages. This was characterized by a drop in pollution-tolerant species and an increase in intolerant species. Our findings indicate that the fish assemblages have improved in Buck Creek over space and time.

INTRODUCTION

The assessment of fish assemblages by management agencies is typically conducted once a year and at one or few sites, based on funding, schedules, and weather. However, fish assemblages vary spatially and temporally due to natural and anthropogenic factors including agriculture, surface run-off, and deforestation (Allan 2004). Assemblages differ between rural and urban watershed land-use types in accordance with abiotic factors such as water temperature, sedimentation, and habitat availability (Falke and Fausch 2010). In addition, fishes require multiple habitats to complete life cycles including spawning, growth, and refuge (Falke and Fausch 2010). Access to habitats can be limited, depending on the fish species and time of year (Roy and Le Pichon 2017).

Lotic environments are excellent systems to study due to the environmental variation that occurs (Grossman and Sabo 2010). Lotic systems that experience high flow variability are typically dominated by generalist species (Poff et al. 2006). Streams with decreased disturbances such as low flow variability are predicted to be dominated by specialists (Poff and Allan 1995). Spring-fed streams are an example of a low disturbance ecosystem, based on low discharge variability that might be dominated by specialist species. In addition, fish assemblages of spring-fed streams frequently vary along the upstream-downstream gradient (Herbert and Gelwick 2003). Stream volume increases with downstream distance, further complicating disturbance patterns with biota, and species richness of fish assemblages increases with stream size (Grenouille et al. 2004, Xenopoulos and Lodge 2006, Roberts and Hitt 2010).

Human activities on the landscape that have consequences for stream environments include agriculture and urbanization (Infante and Allan 2010). Agricultural land-use is a threat to stream ecosystems (Allan 2004). Tile-drained, row crop agriculture results in hydrologic alterations

(Pyron and Neumann 2008) with increased input of pollutants and sediments into streams (Schilling and Helmers 2008). Row-crop agriculture management additionally promotes altered riparian vegetation (Allan 2004). Stream bank vegetation further contributes to in-stream temperature variation (Johnson 2004, Carlson et al. 2014). Rutherford et al. (1997) found that the removal of riparian vegetation results in increased stream temperatures. Knowledge of how stream temperature responds to riparian shading can improve best management practices or restoration (Johnson 2004). Urbanization is an additional land-use extreme, that produces higher surface runoff, peak flow magnitude increase, and water quality degradation (Rose and Peters 2001, Wang et al. 2001). Urbanized streams have increased pollution concentrations and decreased riparian connectedness (Violin et al. 2011).

Effective evaluation of fish assemblages is improved with long-term data (Poff and Allan 1995). Matthews and Marsh-Matthews (2017) described how long-term datasets for fish assemblages have become more available within recent decades. Ecological processes often require years to complete (Franklin 1989) and stream fish assemblages have high temporal variation. A lack of long-term data limits the understanding of mechanisms that drive biodiversity loss in freshwater ecosystems (Jeppesen et al. 2012).

The objectives of this study were to (1) evaluate spatial and temporal variation in the fish assemblages from 1986 to 2018 in Buck Creek, Indiana and (2) demonstrate the value of a long-term dataset. We classified fishes by taxonomic names, trophic traits, pollution tolerance classifications, and analyzed subsequent assemblages for variation that was correlated with environmental variables. We initially hypothesized that fish assemblages would differ predominately with the upstream-downstream gradient. Upstream assemblages are expected to be nested sub-sets of downstream assemblages and composed of habitat or headwater specialists;

downstream assemblages are expected to be dominated by large-river species (Taylor and Warren 2001). We predicted that the implementation of the 1972 Clean Water Act, would shift fish assemblages from being mainly pollution-tolerant species to more intolerant species. Finally, we tested if fish assemblages varied due to in-stream temperature differences and stream bank shading.

METHODS

Study Area

The study was performed on Buck Creek in east-central Indiana. Buck Creek is a mid-sized stream, that flows 37.7 km through Henry and Delaware Counties (Figure 1). It has a mean channel width of 10 m and a drainage area of 259 km^2. The system is a spring-fed, cool-water tributary of the West Fork White River in Muncie, Indiana. The watershed is dominated by row crop agriculture (72%) and urbanization (15%) (USDA 2011). Riparian stream banks are dominated by woody vegetation with scattered grassy strips installed by landowners to manage agriculture runoff.

Field sampling and data analysis

Fishes were sampled annually from 1986-2018 by the Muncie Sanitary District's Bureau of Water Quality (BWQ) at 19 sites in Buck Creek in Delaware and Henry County, IN. For this study, we focused on fish data that were collected by tote-barge electrofishing (Holloway 2018). Field sampling was performed when site turbidity was <40 Nephelometric Turbidity Units.

One tote-barge site was removed due to having one completed sample. Species that were collected only once in the period or were identified to family (not species) were removed from analyses. All analyses used data converted into catch per unit effort (CPUE) by site distance. Annual species turnover rates were calculated in the *codyn* package in R with the *turnover* function. Year-to-year species turnover can mask assemblage composition when measured by species richness alone (Collins et al. 2008; Cleland et al. 2013). We combined focal and previous year observations for proportional species turnover calculated as ([number of species gained] + [number of species lost]) / (total number of species) (Rusch and van der Maarel 1992; Cleland et al. 2013). We confirmed species turnover rates as coefficients of variation (CV/CVs) for all species in all samples. Use of coefficients of variation provides a robust estimation of stability for populations/assemblages (Grossman et al. 1990; Matthews 1998). Lower CV values indicate greater stability for the assemblage, whereas higher values indicate assemblages that are less stable. We used simple linear regression analysis to determine if CV varied with year.

We used nonmetric multidimensional scaling (NMDS) in RStudio (R Core Team, 2019) to ordinate fish assemblages using the *vegan* package version 2.5-5 (Oksanen et al. 2019, ordiellipse and anosim functions). We used Bray-Curtis distances in NMDS and reduced the final solution to a two-dimensional configuration. Ordination plots were visually examined for assemblage variation among sites along the upstream-downstream gradient and years. NMDS is a useful tool for graphical representation of large ecological datasets (Kenkel and Orlóci 1986). Analysis of similarity (ANOSIM) was used to test our hypotheses from the NMDS ordinations. ANOSIM compares mean dissimilarities between groups to mean dissimilarities within the groups (Clarke 1993). CPUE data were log (x + 1) transformed for all NMDS ordinations.

Fish species were categorized by trophic classification (Poff and Allan 1995). Feeding behavior for adult fishes of Buck Creek were from Simon (2011). Tolerance classifications were scored from Simon and Dufour (1998) and tested. We utilized relative abundances of CPUE data for both trophic guild and tolerance analyses.

Rainfall data for Delaware Co., IN were obtained from the National Oceanic and Atmospheric Association (NOAA) from April 1986 through September 2018 (https://www.ncdc.noaa.gov/cag/county/time-series/IN-035/pcp/1/4/1986-2018). Rainfall was predicted to influence stream temperature of Buck Creek (Subehi et al. 2010). Stream bank shading was tested with time to determine if it was related to in-stream temperature. Shading was manually analyzed in ArcGIS Pro. A buffer of 12 m was generated along Buck Creek, with 12 m wide transect lines placed every 30.5 m (Appendix C). Shading was given a value of 0 (no shading), 1 (one bank was shaded), or 2 (both banks were shaded). Available aerial imagery of Delaware County was overlaid, and evaluated, for the years 1994, 1998, 2003, 2005, 2006, 2007, 2008, 2010, 2012, 2014, 2015, and 2016. Once shading evaluation was scored, each year class was summed for a cumulative score. Scores were examined for temporal variation by year with nonparametric correlations.

We utilized a linear mixed effects model to evaluate species richness, CPUE, trophic guilds, tolerance traits, and in-stream temperature over space and time. Because sites were visited each year and the sites are close in proximity to one another (closest sites were 0.5 km apart), each site was treated as a random effect to account for pseudoreplication induced by location. Cohen's d was calculated for effect size of each linear mixed effects model. All analyses were performed in RStudio environment version 1.2.5033 (R Core Team, 2019). Linear mixed effects model used the lme4 package version 1.1-21 (Bates et al. 2019).

RESULTS

The dataset consisted of 32 years of collections at 15 sites from 1986-2018 (Figure 1). A total of 52,213 individuals from 49 species were collected during 205 sampling events (Appendix A). The most abundant family of fish from Buck Creek were Cyprinidae (31%) with 18 species. The most abundant species was *Cottus bairdii* with relative abundance of 29%. According to mixed effects models analysis, species richness increased ($F_{33,171} = 14.44$, $p < 0.001$, $d = 0.35$) with space and time (slope = 0.07, $p = 0.03$, Figure 2). Catch per unit effort decreased ($F_{33,171} = 5.67$, $p < 0.001$, $d = -0.68$) with space and time (slope = -0.02, $p = 0.05$, Figure 3). Annual turnover rate of species in the assemblages decreased ($F_{1,28} = 16.27$, $p < 0.001$) with time ($r = 0.35$, $p < 0.001$, Figure 4). Annual coefficient of variation for fishes of Buck Creek increased ($F_{1,27} = 22.28$, $p < 0.001$) with time ($r = 0.43$, $p < 0.001$, Figure 5).

The spatial NMDS analysis suggested that upstream site fish assemblages (km 20.1-23.9) were distinctly different from downstream site fish assemblages (km 0.3-1.4), and middle site fish assemblages (km 4.9-18.2) ordinated by group (stress = 0.13, Figure 6). The ANOSIM test revealed a difference among the fish assemblages of the sites ($R = 0.54$, $p < 0.001$). A reduction in CPUE for Least Brook Lamprey (*Lampetra aepyptera*) and increase in Black Redhorse (*Moxostoma duquesnei*) and River Chub (*Nocomis micropogon*) with downstream distance was summarized by the spatial NMDS (Figure 6). The NMDS analysis for annual samples suggested early period fish assemblages (1986-1998) were distinctly different from late period fish assemblages (2010-2018), and middle period fish assemblages (1999-2009) plotted within these groups (stress = 0.13, Figure 7). The ANOSIM test revealed differences among the fish assemblages of the annual samples ($R = 0.18$, $p < 0.001$). There was a decrease in Common Carp

(*Cyprinus carpio*) CPUE and an increase in Golden Redhorse (*Moxostoma erythrurum*) and Rock Bass (*Ambloplites rupestris*) from early to late years, respectively (Figure 7).

We found 12 pollution tolerant species with an average relative abundance of 42.9% and 18 pollution intolerant species with an average relative abundance of 16.1% (Appendix B). Pollution tolerant species relative abundance decreased ($F_{30,174} = 29.21$, $p < 0.001$, $d = 1.45$) with space and time ($r = 0.34$, $p < 0.001$, Fig. 8A). The farthest upstream site (km 23.9) had the highest y-intercept, indicating the fish assemblages had more pollution tolerant species. However, all sites showed a decrease in pollution tolerant species with time. Intolerant species relative abundance increased ($F_{30,174} = 29.21$, $p < 0.001$, $d = -1.42$) with space and time (slope = 0.01, $p < 0.001$, Fig. 8B). The most upstream site (km 23.9) had the lowest y-intercept, indicating the fish assemblages had fewer pollution intolerant species. However, all sites demonstrated an increased relative abundance of intolerant species with time. We identified four trophic guilds: herbivore-detritivore, invertivore, omnivore, and piscivore. Spatially, mean invertivore relative abundance was 75%, and mean omnivore relative abundance was 18%. Invertivores and omnivores were temporally dominant too, with invertivores at 60% mean relative abundance, and omnivores at 31% mean relative abundance. Relative abundance of herbivore-detritivores decreased ($F_{30,174} = 1.86$, $p = 0.007$, $d = -0.35$) with space and time (slope = -0.01, $p = 0.02$, Figure 9A). Invertivore relative abundance increased ($F_{30,174} = 2.26$, $p < 0.001$, $d = 0.12$) with space and time (slope = 0.02, $p = 0.01$, Figure 9B). Omnivore relative abundance did not vary ($F_{30,174} = 0.4$, $p = 0.98$, $d = 0.01$) with space and time (slope = 0, $p = 0.6$, Figure 9C). Piscivore relative abundance decreased ($F_{30,174} = 8.48$, $p < 0.001$, $d = -0.23$) with space and time (slope = -0.01, $p = 0.05$, Fig. 9D). In-stream temperature decreased ($F_{30,174} = 5.21$, $p < 0.001$, $d =$

-1.23) with space and time (slope = -0.16, $p < 0.001$, Figure 10). Rainfall for Delaware County did not vary with time. Stream bank shading along Buck Creek increased with time ($r = 0.84$)

DISCUSSION

We observed large changes in the fish community structure of Buck Creek in Delaware County, IN during a 32-year period. Assemblages differed along the upstream-downstream gradient and with time. Spatial variation may be a response to decreased water temperatures. We suggest that significant spatial and temporal trends in water temperature (Figs. 6 and 7) were a result in land management practices and water quality. Multiple fish species and functional traits differed in relative abundance along the longitudinal gradient. This study found that Buck Creek is a cyprinid-dominated system. We found that upstream reaches of Buck Creek were driven by habitat-specific species, while downstream reaches were driven by large-river species. For example, the Least Brook Lamprey, *Lampetra aepyptera,* require clean, flowing headwater streams for spawning and other life history processes (Rice and Zimmerman 2019).

Trajectories of spatial change in fish assemblages of Buck Creek were gradual and directional. Site assemblage changes resulted in a directional shift in the ordination (Fig. 6). Temporal change in Buck Creek fish assemblages were also gradual and directional. Assemblage changes resulted in a leftward shift on the ordination (Fig. 7). Pyron and Deegan (in review) identified similar temporal changes in fish assemblages that they identified as saltatory and either non-directional or directional (as defined by Matthews 1998) within the St. Joseph River of Elkhart and South Bend, Indiana. Spatial fish assemblage variation in Buck Creek was correlated with stream size and habitat availability, in addition to spatial variation in water temperature.

Holloway (2018) found increased Index of Biological Integrity scores for Buck Creek along the upstream-downstream gradient using recent data. We confirmed that the long-term fish assemblage quality in Buck Creek increased significantly with space and time; the number of sensitive species increased with downstream distance. Reash and Berra (1987) found a similar pattern in two Ohio streams, where pollution intolerant species increased with downstream distance. Figs. 8a and 8b depict these improvement patterns as tolerant and intolerant fishes over space and time. Holloway et al. (2018) observed fish assemblages shifting from pollution-tolerant species to sensitive species in a long-term study of the West Fork White River, Indiana. McClelland et al. (2012) found that sensitive and state-threatened species have increased within the Illinois River since the 1990s. We found that the Buck Creek fish assemblages during this period have changed, with higher CPUE of invertivores and decreased CPUE of omnivores.

During this period, in-stream temperature of Buck Creek decreased by an average of 2° C. In-stream temperature increased along the upstream-downstream gradient. We tested rainfall of Delaware County, IN, and riparian shading as potential drivers for the overall decrease in stream temperature. Rainfall patterns for Delaware County over the past 32-years were consistent. Aerial image analysis showed an in increased stream bank shading along Buck Creek during this time. We found that shading varied spatially, but there was an overall decrease from upstream to downstream. This pattern coupled with decreased groundwater input may explain the increase in in-stream temperature with downstream distance.

Conservation reserve programs (CRP) were initiated in 1985 to allow the Farm Service Agency of the USDA to pay farmers for establishing long-term restoration areas (https://www.fsa.usda.gov/programs-and-services/conservation-programs/conservation-reserve-program/, 2019). The increased stream bank shading we observed may be a result from CRP in the Buck Creek watershed. Metzke and Hinz (2017) implemented a stream monitoring program for the Kaskaskia River Basin in Illinois to assess effectiveness of these conservation reserve areas. Metzke and Hinz (2017) reported that CRP/CREP land resulted in only small effects on Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies) assemblages. Kalaninova et al. (2014) found that stream bank shading regulated water temperature and sensitive caddisfly communities. We suggest that CRPs in Buck Creek likely had little effect on the macroinvertebrate assemblage.

Long-term datasets can be an effective asset in evaluating changes to ecological communities and underlying mechanisms (Franklin 1989). Smith et al. (2018) found that both water quality and aquatic macroinvertebrate communities improved following the Clean Water Act. Pyron et al. (2019) found modifications in Ohio River fish assemblages and changes in land-use over 57 years. A similar long-term dataset for the West Fork White River, Indiana resulted in fish body size and geographic range not explaining fish assemblage variation (Jacquemin and Doll 2014). Using a long-term, historical dataset for Ontario lakes Finigan et al. (2018) found that fish communities shifted from cyprinid-dominated to centrarchid-dominated. Hughes et al. (2017) found the scientific community valuing long-term studies more highly than short-term studies. Long-term studies have a large influence of informing environmental policies (Hughes et al. 2017).

In summary, Buck Creek, Indiana fish communities appear to be improving, likely due to increased water quality and vegetated riparian zones. We recommend further conservation efforts including increased riparian vegetation coverage at downstream sites and other best management practices. Similar patterns are likely present for stream fish assemblages elsewhere. Long-term datasets, like the one used here, tell a story focused on the community, and allow local scientists/ managers to see if their current practices are effective or need to be changed.

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DATA ACCESSIBILITY

- Buck Creek site assemblages
- Final Buck Creek site assemblages uploaded as online
- DeRolf_Buck_Creek input file: Dryad doi:10.5061/dryad.m905qfv03
- Sampling location, river mile, species, trophic guild, count, and date: Dryad doi:10.5061/ dryad.m905qfv03

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FIGURE LEGENDS

Figure 1. Sample sites on Buck Creek in Delaware County, Indiana, USA.

Figure 2. Results of linear mixed effects model (random slope and intercept for each sample site) predicting species richness for fish assemblages of Buck Creek, Indiana. Lines correspond to model predictions by sample site.

Figure 3. Results of linear mixed effects model (random slope and intercept for each sample site) predicting catch per unit effort for fish assemblages of Buck Creek, Indiana. Lines correspond to model predictions by sample site.

Figure 4. Annual turnover rate of fish assemblages in Buck Creek 1986-2018

Figure 5. Mean coefficients of variation of CPUE for all species by year for Buck Creek

Figure 6. Non-metric multidimensional scaling biplot of fish assemblages by site in Buck Creek from 1986-2018

Figure 7. Non-metric multidimensional scaling biplot of temporal trends in annual fish assemblages for Buck Creek from 1986-2018.

Figure 8. Results for linear mixed effects model (random slop and intercept by sample site) predicting relative abundance of tolerant (A) and intolerant (B) fishes of Buck Creek, Indiana. Lines correspond to model predictions by sample site.

Figure 9. Results of linear mixed effects model (random slope and intercept for each sample site) predicting relative abundance for trophic guilds of fish assemblages in Buck Creek, Indiana. Lines correspond to model predictions by sample site. Herbivore-detritivores (A), invertivores (B), omnivores (C), and piscivores (D).

Figure 10. Results of linear mixed effects model (random slope and intercept for each sample site) predicting in-stream temperature of Buck Creek, Indiana. Lines correspond to model predictions by sample site.

APPENDICES

Appendix 1: List of Species Collected From 1986-2018

Catostomidae (Suckers)	
Carpiodes carpio	Quillback Carpsucker
Catostomus commersoni	White Sucker
Hypentelium nigricans	Northern Hogsucker
Minytrema melanops	Spotted Sucker
Moxostoma duquesnei	Black Redhorse
Moxostoma erythrurum	Golden Redhorse
Centrarchidae (Sunfishes)	
Ambloplites rupestris	Rockbass
Centrarchidae Family	Hybrid Sunfish
Lepomis cyanellus	Green Sunfish
Lepomis gibbosus	Pumpkinseed
Lepomis macrochirus	Bluegill
Lepomis megalotis	Longear Sunfish
Lepomis microlophus	Redear Sunfish
Micropterus dolomieu	Smallmouth Bass
Micropterus salmoides	Largemouth Bass
Pomoxis annularis	White Crappie
Pomoxis nigromaclatus	Black Crappie
Cottidae (Sculpin)	
Cottus bairdii	Mottled Sculpin
Cyprinidae (Minnows)	
Campostoma anomalum	Central Stoneroller
Cyprinella spiloptera	Spotfin Shiner
Cyprinella whipplei	Steelcolor Shiner
Cyprinus carpio	Common Carp
Luxilus crysocephalus	Striped Shiner
Lythrurus umbratilis	Redfin Shiner
Nocomis biggutatus	Hornyhead Chub
Nocomis micropogon	River Chub
Notemigonus crysoleucas	Golden Shiner
Notropis buccatus	Silverjaw Minnow
Notropis photogenis	Silver Shiner
Notropis rubellus	Rosyface Shiner
Notropis stramineus	Sand Shiner
Notropis volucellus	Mimic Shiner
Pimephales notatus	Bluntnose Minnow
Pimephales promelas	Fathead Minnow
Rhinichthys obtusus	Western Blacknose Dace
Semotilus atromaculatis	Creek Chub
Esocidae (Pikes)	
Esox americanus	Grass Pickerel
Gasterosteidae (Sticklebacks)	
Culaea inconstans	Brook Stickleback
Ictaluridae (Catfishes)	
Ameiurus melas	Black Bullhead
Ameiurus natalis	Yellow Bullhead
Ictalurus punctatus	Channel Catfish
Noturus flavus	Stonecat
Percidae (Perches)	
Etheostoma blennioides	Greenside Darter
Etheostoma caeruleum	Rainbow Darter
Etheostoma nigrum	Johnny Darter
Etheostoma spectabile	Orangethroat Darter
Percina caprodes	Logperch
Percina maculata	Blackside Darter
Petromyzontidae (Lampreys)	
Lampetra aepyptera	Least Brook Lamprey

Appendix 2: Pollution Tolerant and Intolerant Species

Pollution Tolerant					
Black Bullhead	*Amieurus melas*	Common Carp	*Cyprinus carpio*	Green Sunfish	*Lepomis cyanellus*
Blacknose Dace	*Rhinichthys atratulus*	Creek Chub	*Semotilus atromaculatus*	Quillback	*Carpiodes carpio*
Bluntnose Minnow	*Pimephales notatus*	Fathead Minnow	*Pimephales promelas*	White Sucker	*Catostomus commersonii*
Channel Catfish	*Ictalurus punctatus*	Golden Shiner	*Notemigonus crysoleucas*	Yellow Bullhead	*Amieurus natalis*
Pollution Intolerant					
Black Redhorse	*Moxostoma duquesnei*	Longear Sunfish	*Lepomis megalotis*	Rosyface Shiner	*Notropis rubellus*
Golden Redhorse	*Moxostoma erythrurum*	Mimic Shiner	*Notropis volucellus*	Sand Shiner	*Notropis stramineus*
Greenside Darter	*Etheostoma blenniodes*	Northern Hogsucker	*Hypentelium nigricans*	Silver Shiner	*Notropis photogenis*
Hornyhead Chub	*Nocomis biggutatus*	Rainbow Darter	*Etheostoma caeruleum*	Smallmouth Bass	*Micropterus dolomieui*
Least Brook Lamprey	*Lampetra aepyptera*	River Chub	*Nocomis micropogon*	Stonecat	*Noturus flavus*
Logperch	*Percina caprodes*	Rockbass	*Ambloplites rupestris*		

Appendix 3: Example of stream bank shading analysis for Buck Creek